JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION

AMERICAN WATER RESOURCES ASSOCIATION

URBAN IMPACTS ON STREAMS ARE SCALE-DEPENDENT WITH NONLINEAR INFLUENCES ON THEIR PHYSICAL AND BIOTIC RECOVERY IN VERMONT, UNITED STATES[1]

Evan P. Fitzgerald, William B. Bowden, Samuel P. Parker, and Michael L. Kline[2]

ABSTRACT: The physical and biological conditions of stream reaches in 16 watersheds within the Lake Champlain Basin of Vermont, United States, were assessed and analyzed for a response to total impervious area (TIA) at multiple spatial scales. Natural gradients (e.g., channel slope) and human impacts to channel boundary conditions (e.g., bank armoring) were considered to ensure a robust test of the Impervious Cover Model for upslope TIA. The response of geomorphic stability and sensitive macroinvertebrates to TIA was nonlinear and significant ($p < 0.001$), decreasing rapidly at 5% TIA. The effect of urbanization on stream condition was shown to interact significantly with drainage area and channel slope using the analysis of covariance (ANCOVA) ($p < 0.05$). Hydraulic geometry regressions for urban and rural watersheds and ANCOVA were used to describe a significant watershed scale-dependent response of channel width to urbanization ($p = 0.001$). The analysis of macroinvertebrate data from reaches in different stages of channel evolution indicated that stable reaches supported greater richness of pollution intolerant species ($p < 0.001$) and overall taxa richness ($p < 0.01$) than unstable reaches, and that biotic integrity improves as channels regain stability during their evolution into a state of quasi-equilibrium. We conclude that macroinvertebrate communities can respond positively to channel evolution processes leading to natural channel restabilization.

(KEY TERMS: urbanization; watershed management; fluvial processes; restoration; invertebrates.)

Fitzgerald, Evan P., William B. Bowden, Samuel P. Parker, and Michael L. Kline, 2012. Urban Impacts on Streams Are Scale-Dependent with Nonlinear Influences on Their Physical and Biotic Recovery in Vermont, United States. *Journal of the American Water Resources Association* (JAWRA) 1-19. DOI: 10.1111/j.1752-1688.2012.00639.x

INTRODUCTION

The general mechanism for how urban land cover alters the hydrologic cycle of watersheds is well understood (Paul and Meyer, 2001; Wenger *et al.*, 2009). The conversion of natural, pervious surfaces to impervious cover (IC) leads to changes in the timing, duration, and magnitude of streamflows (Leopold, 1968). The physical response of watershed and channel processes to IC often leads to a decrease in drainage density (Dunne and Leopold, 1978), an alteration

[1]Paper No. JAWRA-11-0025-P of the *Journal of the American Water Resources Association* (JAWRA). Received February 22, 2011; accepted January 4, 2012. **Discussions are open until six months from print publication.**

[2]Respectively, Principal Watershed Scientist, Fitzgerald Environmental Associates, LLC, 316 River Road, Colchester, Vermont 05446; Patrick Professor of Watershed Science and Planning, Rubenstein School of Environment and Natural Resources, University of Vermont, Burlington, Vermont 05405; Ecologist, URS Corporation, Fort Washington, Pennsylvania 19034; and Program Manager, Rivers Program, Vermont Department of Environmental Conservation, Waterbury, Vermont 05671 (E-Mail/Fitzgerald: evan@fitzgeraldenvironmental.com).

of geomorphic structure and physical habitat of the channel (Hammer, 1972; Pizzuto *et al.*, 2000), an increase in pollutant conveyance to the channel (Coles *et al.*, 2004), and a decline in biotic richness and diversity both within the channel (Morley and Karr, 2002; Fitzpatrick *et al.*, 2004; Walsh *et al.*, 2005a) and in the adjacent stream corridor (Smith *et al.*, 2009). The sum of urban impacts on stream ecosystems is commonly described as the "urban stream syndrome" (Meyer *et al.*, 2005; Walsh *et al.*, 2005b), and the effects of this syndrome have been summarized across ecoregions in the conceptual Impervious Cover Model (ICM) proposed by Schueler (1994).

Schueler's (1994) conceptual ICM was developed to describe the general response of biotic and abiotic characteristics of stream ecosystems to IC across ecoregions. A recent and extensive review by the Center for Watershed Protection (CWP, 2003) has shown that a response can be detected when the total impervious area (TIA) is at or above 10% of the watershed area (CWP, 2003). The extensive body of literature supporting the ICM is useful because it provides a direct causal relationship between urbanization and degradation of stream ecosystems. Given the public policy implications of the ICM (e.g., zoning regulations to control urbanization, stormwater and site design standards, river restoration, and mitigation plans), there has been considerable interest in whether a threshold exists in the health of stream ecosystems in response to urbanization or whether there is a gradient of decline that is detectable with low levels of TIA (Booth, 2005; Walsh *et al.*, 2005a; Cuffney *et al.*, 2010). A growing body of research across ecoregions indicates that impacts to stream ecosystems are detectable even when TIA is <10%; for example, in the Pacific Northwest (Booth *et al.*, 2002), in the Central United States (U.S.) (Stepenuck *et al.*, 2002; Wang and Kanehl, 2003), and in New England (Morse, 2001; Schiff and Benoit, 2007).

Although a large body of literature supports our understanding of this syndrome and validates the general ICM framework, greater understanding of the specific mechanisms underlying the syndrome is needed to improve the management of urban stream ecosystems (Roy *et al.*, 2009; Wenger *et al.*, 2009). In addition, it is critical to demonstrate that watershed and stream corridor management does, in fact, lead to the improvement of urban stream ecosystems. Numerous studies have highlighted the importance of quantifying the impacts of urbanization on stream ecosystems at different spatial and temporal scales (CWP, 2003; Wenger *et al.*, 2009). In addition to the single stressor-response ICM, we propose the use of additional models to better understand the scaling of impacts caused by urbanization, the interactions between natural and anthropogenic landscape features, and the recovery potential of biotic communities in urban watersheds. These models include downstream hydraulic geometry (DHG) for urban and rural watersheds, and channel evolution stages and associated biological indices.

DHG regressions were originally developed by Leopold and Maddock (1953) to describe increases in channel dimensions in a downstream direction along the channel network. In self-formed alluvial channels, bankfull channel dimensions are predicted using the following power equation:

$$\mathrm{CD} = \alpha Q^{\beta}, \qquad (1)$$

where CD is the bankfull channel geometry (m or m^2), Q is the bankfull discharge (m^3/s), α is the regression coefficient, and β the regression exponent. Drainage area (DA) (km^2) is often used as a surrogate for bankfull discharge in its absence. Although DHG regressions have been explored across different spatial scales (Wohl, 2004) and vegetation types (Anderson *et al.*, 2004), only a few DHG regressions have been developed and explored for urban watersheds (e.g., Allen and Narramore, 1985; Neller, 1989; Doll *et al.*, 2002; Hession *et al.*, 2003). DHG regressions have been promoted by many federal and state governmental agencies in the U.S. as an essential design tool for stream corridor restoration projects (Rosgen, 1996), including projects in urban watersheds. However, some researchers have questioned the utility of DHG regressions within a process-based approach to channel restoration (Simon *et al.*, 2007), and have called for a better understanding of the natural and human-induced processes influencing channel stability. DHG regressions that compare urban and rural watersheds from the same physiographic region add to our understanding of the different natural and anthropogenic effects on stream channel dimensions and processes, and how these effects vary and interact across watershed scales.

Channel evolution models provide a framework for understanding and predicting channel adjustments triggered by watershed and reach scale human disturbances. When a stream channel's capacity to transport sediment is equivalent to its sediment supply, the channel is said to be in a state of dynamic equilibrium (i.e., force equals resistance; Lane, 1955). However, changes in a watershed's hydrologic regime caused by urbanization can lead to significant departures in stream channel morphology and processes (Schumm, 1999). The evolutionary sequence of channel adjustments from stable, equilibrium conditions to unstable states has been characterized by various researchers (Schumm *et al.*, 1984; Simon, 1989). Although channel evolution sequences vary slightly by author, they have in common four process-based adjustments: degradation (incision), widening, planform changes (lateral adjustments), and aggradation. Understanding the channel evolution sequence in urbanizing watersheds is critical for future predictions of channel form, function (Bledsoe *et al.*, 2002), and biotic integrity during destabilization and recovery to a state of quasi-equilibrium. The characterization of channel evolution stage in stream reaches allows for a space-for-time substitution to interpret the pattern and trajectory of channel adjustments and the response of biotic communities. To date, little research has explored these linkages and therefore our understanding of biological recovery potential in urban watersheds is very limited.

In this study, we explored the scaling of urban impacts on stream conditions in watersheds within the Lake Champlain Basin of Vermont by testing: (1) the effect of urbanization on geomorphic stability, physical habitat conditions, and biotic communities at three different spatial scales; (2) the differences between urban and rural DHG; and (3) the response of biotic communities to channel evolution processes. We expected that geomorphic stability and macroinvertebrate richness would decline as TIA increased and that TIA measured at the watershed scale would better predict stream conditions than TIA measured at local scales. We hypothesized that DHG regressions would be better developed (i.e., higher coefficient of determination) in rural watersheds than urban watersheds. Finally, we hypothesized that stream reaches that had re-established geomorphic stability after a period of destabilization caused by human impacts would support greater macroinvertebrate species richness than unstable reaches and would support levels of species richness similar to the predisturbed condition.

METHODS

Study Area

In this study, we use the term "urban" to describe watersheds that have been identified as biologically impaired by stormwater runoff by the Vermont Department of Environmental Conservation (VTDEC). As part of a collaborative effort between the University of Vermont (UVM) and VTDEC, we collected reach-scale geomorphic stability and physical habitat data within 11 urban watersheds in northwestern Vermont. An additional five rural watersheds that currently meet the VTDEC water-quality standards were selected for comparison to the urban watersheds. The total set of 16 rural and urban watersheds represent a continuum of urbanization characteristics (e.g., percent TIA, percent forest). Macroinvertebrate data were collected by VTDEC and UVM in 13 of the 16 study watersheds to further analyze the effects of TIA and geomorphic stability on biotic integrity.

The 16 study watersheds are located in the Lake Champlain Basin in northwestern Vermont (Figure 1). The postglacial topography of the area is characterized by rolling hills with mean watershed elevations ranging from 46 to 255 m above mean sea level. The surficial geology is dominated by glacial till soils overlain by silts and clays deposited during the early Holocene when the entire valley was occupied by the Champlain Sea (Wright, 2003). The land use in the urban watersheds is characterized by a mix of residential, commercial, industrial, agricultural, and forested cover types. The rural watersheds are located beyond the fringe of urban development surrounding the cities of Burlington and St. Albans in landscapes with a mixture of forested and agricultural land uses and with minimal amounts of low-density residential land use. Nearly all of the forested lands in the Lake Champlain Basin were cleared for agriculture during the mid to late 1800s, and

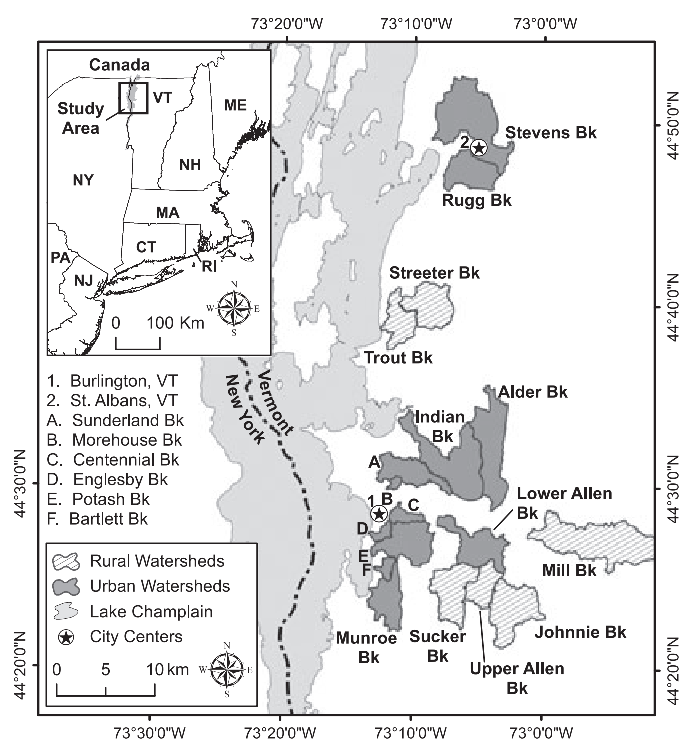

FIGURE 1. Site Map of Study Watersheds in Northwestern Vermont, U.S.

present-day forested areas are typically second and third growth stands (Thompson and Sorenson, 2000).

Spatial Analysis of Total Impervious Area

Geographic information system (GIS) software (ESRI's ArcGIS®, ESRI, Redlands, CA) was used to delineate and quantify watershed areas and land use at three different spatial scales: (1) the complete upslope DA (upslope area) to the reach, (2) the local subwatershed area draining directly to the reach (local area), and (3) the stream corridor (corridor) defined as the area within 30 m of either side of the stream channel. For statistical analyses using the percent TIA of the upslope area as one independent variable, TIA was measured for the DA beginning at the downstream reach break and extending upslope.

Land use data derived from two separate sources for the study area was utilized to quantify percent TIA for each DA (Fitzgerald, 2007). Statewide Landsat imagery collected in 2002 using a 30-m grid was processed by UVM's Spatial Analysis Laboratory (SAL), resulting in the following four spectral classes: (1) forest, (2) urban, (3) open (agricultural and open recreational uses), and (4) water and other (SAL, 2005). In addition, a separate dataset of TIA derived by Morrissey and Pelletier (2006) from high-resolution Quickbird satellite scenes collected between 2003 and 2005 was utilized. The multispectral bands (2.4 m resolution) from the Quickbird scenes were analyzed by SAL using Definiens eCognition® software to classify the data into three classes: (1) impervious, (2) pervious, and (3) water. Quickbird-derived TIA data were only available for a select group of watersheds during the time of this study. Given this limitation, a correlation analysis was performed using the Landsat-derived urban class and the Quickbird-derived impervious class that resulted in a robust linear relationship ($R^2 = 0.96$), allowing for the calculation of percent TIA for all study watersheds at each spatial scale.

Reach Selection Criteria

To test the ICM in our study area, we used an independent watershed approach to measure the effect of TIA on field measured variables of stream condition. This approach to site selection, similar to that taken by other researchers (Roy *et al.*, 2003; Coles *et al.*, 2004), results in a single site per watershed to test for land use effects on independent reaches. Two of the 16 study watersheds contained more than one independent subwatershed, resulting in a total of 18 possible locations for reach selection. Using this approach, we selected high- and low-gradient reach types from each independent subwatershed. The criteria used for distinguishing between high- and low-gradient stream types are consistent with VTDEC's Stream Geomorphic Assessment (SGA) protocols (VTDEC, 2005) and are based primarily on channel form, bed substrate, and channel slope (Rosgen, 1994; Montgomery and Buffington, 1997). High-gradient reaches are characterized by coarse-grained bed substrate with channel slopes >0.5%, and low-gradient reaches are characterized by fine-grained bed substrate with channel slopes ≤0.5%. Separate ICM analyses of high- and low-gradient stream types were conducted to reduce the gradient of natural characteristics that covary with anthropogenic factors (Allan, 2004).

For both stream types, reaches were screened for selection using the following additional criteria: (1) the reach is located in the most downstream area of the watershed or subwatershed; and (2) the stream corridor has limited legacy effects of channel straightening, berming, or armoring. The first criterion aids in selecting larger-sized (e.g., width) stream reaches having features that are more readily observed than those in lower-order (Strahler, 1964) stream reaches. The second criterion screens reaches with immediate and direct impacts to the channel boundary conditions, allowing for a more specific ICM test of the effect of upslope TIA on the selected reach (e.g., the effect of urbanization at the watershed scale *vs.* the local scale). Using these criteria within the 18 independent watersheds, 17 reaches were suitable for use in the high-gradient analysis and 12 reaches were suitable for use in the low-gradient analysis. Fewer low-gradient reaches were available because some of the smaller watersheds had high relief ratios (>30) (Dunne and Leopold, 1978) with no low-gradient reaches. Macroinvertebrate samples were collected only where coarser bed substrates are suitable for VTDEC biomonitoring protocols (VTDEC, 2006). Therefore, the analysis of biotic response to TIA was limited to the high-gradient dataset for the 17 independent watersheds. A total of 14 reaches within these watersheds were sampled for macroinvertebrates.

Reaches selected for DHG regression analysis included multiple sites per watershed. This approach is consistent with the original DHG methods developed by Leopold and Maddock (1953) and those used in recent analyses in small urban and rural watersheds (Doll *et al.*, 2002; Hession *et al.*, 2003). Reaches were classified as nonalluvial channels and were eliminated if they contained significant bank or bed armoring (natural or anthropogenic) or were located in close proximity to bridges or culverts where human-caused adjustments are prevalent. Separate

DHG regressions were developed for high- and low-gradient stream types.

Reaches used to explore linkages between geomorphic stability and macroinvertebrates also included multiple sites per watershed. For this analysis, the reach selection criteria for completely independent reaches were considered less important as the area of interest was the local scale effect of channel form and process on the biotic community (unlike the ICM test of the effects of upslope watershed TIA). Thirty-six reaches from 15 of the urban and rural study watersheds with available macroinvertebrate and geomorphic assessment data were selected for this analysis. To explore linkages between current geomorphic stability (as assessed in 2005 and 2006) and biotic integrity, we used only VTDEC and UVM macroinvertebrate sampling results collected from 2000 to 2007. Due to substantial urbanization within many of the study watersheds during the past 10 years, samples collected prior to 2000 might reflect biotic communities found under different hydro-geomorphic conditions and were therefore omitted.

Field and Laboratory Methods

The geomorphic stability and physical habitat conditions of stream reaches were surveyed by UVM during 2005 and 2006 using SGA protocols developed by VTDEC (2005). Reaches were initially delineated using remote-sensing techniques and a GIS database that included hydrography, geology, soils, and topography data. Reach breaks were operationally defined along the channel network based on changes in (1) stream confinement (valley width), (2) valley slope, (3) geologic materials, and (4) tributary influences according to the VTDEC SGA protocol (VTDEC, 2005). During field data collection, reach delineations were verified or adjusted if necessary based on direct field observations of changes in valley and channel morphology. In cases where further field segmentation was required due to variability that was not detectable with remotely sensed data, reaches were divided into smaller distances for surveying. A minimum reach length of 100 m was maintained.

The SGA protocol utilizes a combination of quantitative and qualitative measurements to calculate two composite indices: (1) the Rapid Geomorphic Assessment (RGA) score; (2) the Rapid Habitat Assessment (RHA) score. Individual measurements used for RGA characterized the channel and floodplain geometry and planform, bed substrate, bank erosion, and bank and buffer vegetation. Within each reach, one to three cross-sectional profiles were collected at a representative riffle using levels and standard surveying rods. For all reaches used in the development of DHG regressions, between three and four cross-sectional measurements were collected at riffle locations where the thalweg cross-over occurs (i.e., where flow is laminar) to obtain average channel geometry dimensions. Bankfull width, floodprone width, mean depth, maximum depth, and low-bank height were collected at each cross-section to calculate a bankfull width to depth ratio, entrenchment ratio, and incision ratio (Rosgen, 1996; VTDEC, 2005). Bed substrate was characterized by 100 randomly collected measurements across different geomorphic features in each reach using a pebble count methodology adapted from Wolman (1954) and described in the SGA protocols (VTDEC, 2005). Substrate embeddedness was also evaluated during the sampling of bed substrate (Barbour *et al.*, 1999). Large woody debris (LWD) that was at least 15 cm in diameter and 1.5 m in length was tallied for each reach. Sediment bars were evaluated for frequency and type (e.g., side bar). Streambank material cohesiveness was qualitatively evaluated, and the height and length of bank erosion and armoring were measured for each bank. Bank and buffer vegetation was classified based on general categories (e.g., coniferous, deciduous), and canopy cover (percent cover) was visually estimated for each bank.

Stream reaches were classified based on channel form using the Rosgen's (1994) methodology, bed morphology and planform (Montgomery and Buffington, 1997), and channel evolution stage (Schumm *et al.*, 1984; VTDEC, 2005). At channel cross-sections and additional locations along each reach, valley-to-valley observations were made to determine the presence or absence of modern terraces (i.e., those formed by anthropogenically induced channel adjustments *vs.* historical geologic features) indicating an advanced stage of channel evolution and a distinction between Stages I and V. Where available, historical aerial photography from the 1930s and 1970s was reviewed and compared with recent aerial photography from the last 20 years to evaluate changes in channel planform and further interpret the present-day channel evolution stage.

The location of field survey points important for data analysis (e.g., cross-section locations) was recorded using a global positioning system. All field surveys were conducted during base-flow conditions by observers trained during the same VTDEC training session in May 2005.

Using individual SGA field measurements described above, the physical stream condition was quantified by assessing the effect of channel adjustment processes on reach equilibrium conditions (Lane, 1955). The composite RGA calculation uses equal weighting of four adjustment processes

(Table 1) resulting in a score that ranges from 0 to 1.0, with a score of 1.0 reflecting a stable, undisturbed reach in dynamic equilibrium conditions (Lane, 1955).

The RHA is a composite score adapted from the USEPA's Rapid Bioassessment Protocols (Barbour *et al.*, 1999). The RHA combines individual quantitative measurements with a suite of qualitative observations to evaluate the following physical habitat conditions: epifaunal substrate and cover, pool substrate and variability, sediment deposition, channel-flow status, channel armoring, channel sinuosity, bank stability and vegetation, and riparian buffer width. The composite RHA uses equal weighting of 10 parameters to calculate a score that ranges from 0 to 1.0, with a score of 1.0 reflecting reference habitat conditions.

TABLE 1. Stream Channel and Floodplain Characteristics Used to Evaluate Adjustment Processes in Calculation of the Composite Rapid Geomorphic Assessment (RGA) Score.

Adjustment Process 1: Channel degradation (0-20 points)
Exposed till or fresh substrate in the stream bed and exposed infrastructure
New terraces or recently abandoned floodplains (incision ratio >1.4)
Headcuts, or nickpoints that are 2-3 times steeper than typical riffle
Freshly eroded, vertical banks
Alluvial river sediments that are imbricated high in bank
Tributary rejuvenation, observed as nickpoints at or upstream of the mouth of a tributary
Bars with steep faces, usually occurring on the downstream end of a bar
Adjustment Process 2: Channel aggradation (0-20 points)
Shallow pool depths, channel homogeneity
Abundant sediment deposition on point bars and mid-channel bars and extensive sediment deposition at obstructions, channel constrictions, and at the upstream end of meander bends
Channel bed is highly exposed during typical low-flow periods
High frequency of debris jams
Coarse pebbles, cobbles, and boulders embedded with sands, silts, and granules
Adjustment Process 3: Channel widening (0-20 points)
Active undermining of bank vegetation on both sides of the channel; many unstable bank overhangs that have little vegetation holding soils together; high width to depth ratio
Erosion on both right and left banks in riffle sections
Recently exposed tree roots
Fracture lines at the top of the bank that appear as cracks parallel to the river
Mid-channel bars and sidebars may be present
Urbanization and stormwater outfalls leading to increased runoff and channel enlargement
Adjustment Process 4: Change in planform (0-20 points)
Flood chutes or neck cutoffs may be present
Channel avulsions may be evident or impending
Island formation and/or multiple thread channels observed
Thalweg is misaligned and not observed in normal pattern traveling from the outside of a meander bend to the outside of the next meander bend
As a result of the lateral extension of meander bends, additional deposition and scour features may be in a channel length typically occupied by a single riffle-pool sequence

Note: RGA score is calculated by taking the sum of the four adjustment process scores and dividing by 80 to normalize to a range of 0 to 1 (VTDEC, 2005).

Macroinvertebrate samples were collected from 13 of the 16 watersheds by VTDEC and UVM. Samples were collected during September through early November during base-flow conditions from representative riffles using a 500-μm mesh-bottom kicknet (0.14 m^2 sampling area) following VTDEC's (2006) composite riffle-sampling methodology adapted from Barbour *et al.* (1999). Four separate locations within the riffle, representing a range of velocity and substrate types characteristic of the riffle, were sampled by agitating the substrate in a 0.2-m^2 area upstream of the kicknet. Each of the four locations were actively sampled for 30 s (total sample time of 2 min), and the contents collected in the kicknet were preserved in a container with 80% ethanol. Two replicate samples were collected from each riffle at independent locations within the riffle.

In the laboratory, samples were thoroughly washed through a 500-μm mesh brass sieve. Each sample was back-washed into a 30-cm × 45-cm tray delineated with 24 equally sized squares. A random selection technique was used to determine the order of square identification. All organisms were removed from each square before proceeding to the following selected square and identification continued until a minimum of six squares (25% of the sample) had been completed. If <300 organisms had been identified in the first six squares, identification continued until a minimum of 300 organisms was reached upon completion of the final square. Organisms were identified to the species level by VTDEC biologists, and tabulated species data were used to calculate the biotic index (BI) (Hilsenhoff, 1987), overall sample richness, and richness for Ephemeroptera, Plecoptera, and Trichoptera (EPT) orders.

Statistical Analyses

Correlation analysis on ranked data was used to produce a matrix of Spearman's ρ values that allowed for the exploration of collinearity between variables. For all ICM analyses, variables with non-normal distributions were log-transformed to normalize the data for analysis using MINITAB® Statistical Software (Minitab, State College, PA). Single predictor models analyzing the effect of TIA on physical stream conditions and biota were developed using least-squares regression on log-transformed data. A stepwise regression analysis (backward elimination; $p = 0.1$ to remove) was then performed using TIA and other natural watershed characteristics to determine the best-fit multiple linear regression models for each spatial scale of TIA (i.e., upslope area, local area, and corridor) and to determine the relative influence of each predictor variable on stream condition. Due to the small sample sizes for ICM analyses of independent reaches for physical conditions ($n = 17$) and biotic conditions ($n = 14$), multiple linear regression models were restricted to three variables with no interaction terms. However, more complicated effects involving the interaction of urbanization with watershed DA and channel slope were explored using the analysis of covariance (ANCOVA).

DHG regressions were developed using the power law equation approach (Leopold and Maddock, 1953). Data for low-gradient reaches yielded poorly developed DHG relationships (coefficients of determinations <0.5) and were therefore not analyzed in the same detail as the regressions for high-gradient reaches. ANCOVA was used on log-transformed data for high-gradient reaches to test for interaction of watershed DA with the urbanization effect (watershed type) in the response of channel dimensions, following Hession *et al.* (2003). The ANCOVA approach was to test for significant DHG regressions and for evidence of a scale-dependent response of channel dimensions to urbanization.

Boxplots depicting the response of macroinvertebrates to channel evolution stages for incising reaches (Schumm *et al.*, 1984; VTDEC, 2005) were visually evaluated for trends. A nonparametric Mann-Whitney test was used to compare biotic metrics between individual channel evolution stages. Channel evolution stages were also divided into stable and unstable categories and a Mann-Whitney test was performed on the two categories for EPT richness, overall taxa richness, and BI.

RESULTS

Watershed and Reach Characteristics

DA to the study reaches ranged from 0.7 to 41.9 km^2 (Table 2). Channel slopes ranged from 1.0 to 3.6% for the high-gradient dataset and were all below 0.6% for the low-gradient dataset. Mean reach elevations ranged from 29 to 236 m above mean sea level. Watershed TIA ranged from 0.6 to 39.3% across the study watersheds. Values of TIA all exceeded 5% in most of the urban watersheds, whereas values of TIA in the rural watersheds were all <5%. The mean RGA score was 0.75 for rural reaches and 0.53 for urban reaches (low- and high-gradient reaches combined; Table 3). Channel incision ratios were higher in urban reaches (mean = 1.47) than rural reaches (mean = 1.15), indicating a higher degree of floodplain disconnection in the urban watersheds and, in most cases, advanced stages of channel evolution stages (Table 3). DHG regressions for high-gradient reaches were developed using 24 reaches from urban watersheds and 19 reaches from rural watersheds (Table 4).

Impervious Cover Model Analysis

Correlations of ranked data (Table 5) indicated that RGA and RHA scores were highly correlated and therefore a single metric, RGA score, was used to summarize ICM results for physical stream conditions. Similarly, EPT richness and overall taxa richness were significantly correlated, and EPT richness was chosen to summarize ICM results for biota. BI and EPT were negatively correlated, but BI was not significantly correlated with TIA. Upslope percent TIA and percent forest cover were negatively correlated, and because the primary purpose of this study was to determine the effect of urbanization on stream conditions, percent forest cover was excluded from the regression analyses. Channel slope and watershed DA, variables that are typically correlated in many watersheds as slope decreases in a downstream direction (Dunne and Leopold, 1978), were also negatively correlated in our study area.

Single predictor regressions using TIA as a percentage of the upslope area resulted in significant nonlinear models to predict RGA (Figure 2a) and EPT richness (Figure 3) in high-gradient reaches. The single predictor model for RGA in high-gradient reaches (Figure 2a) explained much more variance than the model for low-gradient reaches (Figure 2b). The comparison of these two models shows a steeper declining response in high-gradient RGA (Figure 2a; regression slope = −0.151) than in low-gradient RGA (Figure 2b; regression slope = −0.036) in response to TIA.

Stepwise regression analyses for multiple spatial scales indicated that TIA was a significant variable in all models of RGA (Table 6). For the upslope and local area models, TIA was the only significant variable in the model; however, DA was also a significant variable ($p < 0.1$) in the corridor model. TIA was a significant variable in upslope and local area models of EPT, but not the corridor model. Percent cobble substrate was a significant variable and explained 11% of the variance of EPT in the upslope area model. For the corridor model of EPT, only DA and percent sand substrate were significant predictors.

TABLE 2. Watershed and Stream Characteristics for ICM Study Reaches.

Reach Name	Reach Type	Drainage Area (km^2)	Reach Upslope TIA (%)	Channel Slope (%)	Mean Elevation (m)	Bed Substrate[1]	Channel Bedform[2]	Stream Type[3]
High-gradient reaches								
Allen Bk 1	Urban	29.0	7.0	1.0	63	Fine gravel	Pool-riffle	C
Bartlett Bk 1	Urban	2.7	15.2	3.2	33	Fine gravel	Step-pool	B
Centennial Bk 1	Urban	3.9	29.0	3.3	53	Coarse gravel	Step-pool	B
Englesby Bk 1D	Urban	1.9	21.0	2.1	46	Fine gravel	Plane-bed	C
Indian Bk 9A	Urban	19.5	9.0	2.4	75	Fine gravel	Plane-bed	C
Morehouse Bk 2A	Urban	1.1	32.6	2.3	64	Coarse gravel	Plane-bed	B
Munroe Bk 4B	Urban	6.3	5.7	1.3	59	Fine gravel	Pool-riffle	C
Munroe Bk T1B	Urban	3.2	5.9	1.7	51	Coarse gravel	Plane-bed	C
Potash Bk 2B	Urban	18.1	19.8	1.2	39	Coarse gravel	Pool-riffle	C
Rugg Bk 8A	Urban	6.8	8.1	1.7	127	Fine gravel	Pool-riffle	C
Stevens Bk 7	Urban	4.0	10.8	3.2	128	Fine gravel	Step-pool	B
Allen Bk 6	Rural	10.1	2.7	2.9	170	Cobble	Plane-bed	B
Johnnie Bk 2	Rural	16.6	1.2	3.6	99	Cobble	Step-pool	B
Mill Bk 2	Rural	41.9	1.6	2.8	98	Cobble	Plane-bed	B
Streeter Bk 1	Rural	17.3	3.9	1.4	63	Fine gravel	Pool-riffle	C
Sucker Bk 3	Rural	17.7	2.6	1.3	108	Coarse gravel	Pool-riffle	C
Trout Bk 7	Rural	1.9	3.1	3.5	78	Cobble	Step-pool	B
Low-gradient reaches								
Allen Bk 4B	Urban	18.6	4.1	0.4	134	Sand	Plane-bed	E
Centennial Bk 3	Urban	3.8	31.1	0.5	65	Sand	Dune-ripple	E
Indian Bk 1	Urban	31.3	8.4	0.2	29	Sand	Dune-ripple	E
Morehouse Bk 3	Urban	0.7	39.3	0.5	72	Sand	Dune-ripple	E
Munroe Bk 2B	Urban	9.2	7.0	0.5	50	Sand	Plane-bed	E
Munroe Bk T1A	Urban	4.6	4.9	0.4	46	Sand	Dune-ripple	E
Potash Bk 7	Urban	15.4	20.3	0.4	65	Sand	Dune-ripple	E
Sunderland Bk 3	Urban	10.5	16.1	0.2	43	Sand	Dune-ripple	E
Johnnie Bk 1B	Rural	17.0	1.1	0.5	90	Fine gravel	Pool-riffle	E
Mill Bk 11	Rural	18.6	0.6	0.3	236	Fine gravel	Pool-riffle	E
Sucker Bk 2	Rural	18.7	2.6	0.2	103	Sand	Plane-bed	E
Trout Bk 1	Rural	12.2	1.7	0.3	31	Sand	Plane-bed	E

Notes: TIA, total impervious area.
[1]Median bed substrate: sand (0.06-2mm), fine gravel (2-16 mm), coarse gravel (16-64 mm), and cobble (64-256 mm).
[2]From Montgomery and Buffington (1997).
[3]From Rosgen (1994).

Reach elevation and channel slope were not significant in any of the models developed through the stepwise regression analysis.

Although channel slope was not a significant variable in the stepwise regression analysis, ANCOVA results indicate that both channel slope and DA were significant covariates ($p < 0.05$) in the upslope area model of RGA (Table 7), confirming that variables associated with watershed scale interact with urbanization in its effect on physical stream condition. Neither channel slope nor DA was found to be a significant covariate in the EPT predictor models.

Downstream Hydraulic Geometry Regressions

DHG regressions for high-gradient reaches were developed to analyze the response of channel dimensions to the effect of DA (Figure 4). Regressions for channel width, depth, and cross-sectional area (variables not plotted in Figure 4) were all significant (Table 7) and had R^2 values >0.50. Using ANCOVA, we found a significant effect of urbanization and a significant urbanization-DA interaction for channel width, and for channel cross-sectional area, but not for channel depth (Table 7). Results indicate that there is a scale-dependent response of channel width and cross-sectional area to urban land cover. This effect is more pronounced in lower-order reaches and becomes less pronounced at a DA of $\sim$15 km^2.

Biotic Response to Channel Stability

Geomorphic assessment data and macroinvertebrate sampling data were analyzed for 36 high-gradient stream reaches (Table 3 and Table S1) to explore the response of biota to channel stability. Biotic data for some stages of channel evolution were not

TABLE 3. Reach Geomorphic and Biotic Assessment Results for ICM Study Reaches.

		Reach Channel Metrics[1]				RGA Adjustment Scores[2]									
Reach Name	**Reach Type**	**Sinuosity**	**IR**	**W:D**	**ER**	**Deg.**	**Agg.**	**Wid.**	**Plan.**	**CEM Stage**	**RGA Score**	**RHA Score**	**EPT Richness[3]**	**Overall Richness[3]**	**Hilsenhoff BI[3]**
High-gradient study reaches															
Allen Bk 1	Urban	1.8	1.4	12.4	9.8	11	11	12	11	III	0.56	0.6	6	36	6.5
Bartlett Bk 1	Urban	1.1	1.6	8.5	1.7	7	10	10	11	II	0.48	0.53	6	24	5.9
Centennial Bk 1	Urban	1.2	1.2	14.9	2.3	13	9	8	12	III	0.53	0.59	3	28	6.4
Englesby Bk 1D	Urban	1.2	1.2	18.1	4.0	9	5	8	8	IV	0.38	0.44	6	25	3.4
Indian Bk 9A	Urban	1.1	1.2	19.3	3.5	13	12	11	10	III	0.58	0.64	10	42	5.3
Morehouse Bk 2A	Urban	1.1	1.8	22.0	3.3	4	5	6	7	III	0.28	0.47	5	25	3.6
Munroe Bk 4B	Urban	1.1	1.9	9.1	6.9	9	12	13	13	II	0.59	0.68	9	22	3.0
Munroe Bk T1B	Urban	1.2	1.8	11.6	9.5	11	11	11	11	IV	0.55	0.46	8	29	3.5
Potash Bk 2B	Urban	1.1	1.8	20.3	2.4	9	10	8	9	III	0.45	0.37	7	28	5.6
Rugg Bk 8A	Urban	1.1	1.1	17.5	3.0	13	13	14	10	IV	0.63	0.65	7	34	6.1
Stevens Bk 7	Urban	1.0	1.7	18.9	1.4	6	10	9	11	II	0.45	0.54	8	44	5.2
Allen Bk 6	Rural	1.1	1.0	14.6	1.4	16	15	15	16	I	0.78	0.85	25	68	3.1
Johnnie Bk 2	Rural	1.1	1.0	23.2	1.4	18	16	16	18	I	0.85	0.88	NA	NA	NA
Mill Bk 2	Rural	1.1	1.0	23.0	2.2	14	16	16	17	I	0.79	0.8	27	64	4.0
Streeter Bk 1	Rural	1.2	1.2	12.8	2.2	16	15	14	17	I	0.78	0.9	14	49	4.2
Sucker Bk 3	Rural	1.3	1.4	16.0	4.0	14	15	12	15	III	0.7	0.69	NA	NA	NA
Trout Bk 7	Rural	1.1	1.1	16.2	2.6	17	18	15	19	I	0.86	0.84	NA	NA	NA
Low-gradient study reaches															
Allen Bk 4B	Urban	1.0	1.4	10.7	6.4	10	10	11	12	II	0.54	0.62	NA	NA	NA
Centennial Bk 3	Urban	1.1	1.3	6.7	9.8	12	12	10	13	III	0.59	0.57	NA	NA	NA
Indian Bk 1	Urban	1.5	1.1	8.6	30.7	14	14	15	15	I	0.73	0.71	NA	NA	NA
Morehouse Bk 3	Urban	1.0	1.4	4.3	17.1	9	11	12	10	III	0.53	0.37	NA	NA	NA
Munroe Bk 2B	Urban	1.0	2.3	7.1	1.6	4	9	10	9	II	0.4	0.42	NA	NA	NA
Munroe Bk T1A	Urban	1.1	1.2	6.1	25.8	14	14	13	13	I	0.68	0.58	NA	NA	NA
Potash Bk 7	Urban	1.3	1.4	8.3	3.2	9	11	12	15	II	0.59	0.66	NA	NA	NA
Sunderland Bk 3	Urban	1.1	1.2	8.0	18.6	12	12	11	10	II	0.56	0.5	NA	NA	NA
Johnnie Bk 1B	Rural	1.1	1.2	9.9	15.0	12	13	14	12	II	0.64	0.56	NA	NA	NA
Mill Bk 11	Rural	1.7	1.3	10.6	31.9	13	15	15	15	IV	0.73	0.7	NA	NA	NA
Sucker Bk 2	Rural	1.2	1.2	12.7	12.1	12	13	14	15	I	0.68	0.69	NA	NA	NA
Trout Bk 1	Rural	1.2	1.1	8.5	5.8	14	14	13	13	I	0.68	0.57	NA	NA	NA

Notes: CEM, channel evolution model from Schumm *et al.* (1984); RGA, rapid geomorphic assessment; RHA, rapid habitat assessment; NA, macroinvertebrate data not applicable for ICM analysis.

[1]Channel metrics: IR, incision ratio; W:D, width-to-depth ratio; ER, entrenchment ratio.

[2]Adjustment processes: Deg., degradation; Agg, aggradation; Wid., widening; Plan., planform.

[3]Species richness for Ephemeroptera, Plecoptera, and Trichoptera; overall species richness for entire sample; Hilsenhoff (1987) biotic index.

normally distributed, and sample sizes for some stages were small (Figure 5). Therefore, when comparing BIs between individual channel evolution stages, a nonparametric Mann-Whitney test was utilized. When comparing Stage I with other stages, all three BIs were significantly different from Stage II ($p < 0.0001$ for EPT; $p < 0.01$ for overall richness and BI) and Stage III ($p < 0.0001$ for EPT; $p < 0.05$ for overall richness; $p < 0.01$ for BI). When comparing Stage V with other stages, only EPT and BI were significantly different from Stage II ($p < 0.05$) and Stage III ($p < 0.05$). No other significant differences were measured between groups. The response of EPT richness to channel evolution processes (Figure 5) revealed a pattern of decline beginning with the incision and widening stages (II and III), with a recovery trend through the aggradation (IV) and restabilization (V) stages. A similar trend was noted for the response of taxa richness and BI. Reaches classified as Stages I (equilibrium conditions) and V (quasi-equilibrium conditions) were grouped as stable ($n = 12$), and those classified as Stages II through IV were grouped as unstable ($n = 24$). Results of nonparametric Mann-Whitney tests indicate that stable reaches support greater EPT richness ($p < 0.001$) and overall taxa richness ($p < 0.01$) than unstable reaches, with a significant difference also observed for BI ($p < 0.001$).

DISCUSSION

Impervious Cover Model

Our data are entirely consistent with the overwhelming evidence that supports the stressor-response

TABLE 4. Data Summary for High-Gradient Reaches Used to Develop DHG Regressions.

Watershed	Watershed Type	Reach	Drainage Area (km^2)	Channel Width (m)[1]	Channel Depth (m)[1]	Channel Area (m^2)[1]
Allen Brook	Urban	1	29.1	9.5	0.8	7.6
Allen Brook	Urban	2	27.9	11.8	0.8	9.5
Allen Brook	Urban	3	26.3	9.8	0.6	5.8
Allen Brook	Urban	4	26.3	8.6	0.5	4.2
Allen Brook	Urban	5	13.5	5.8	0.4	2.2
Bartlett Brook	Urban	1	2.7	3.6	0.4	1.5
Bartlett Brook	Urban	2	1.7	4.0	0.5	1.8
Bartlett Brook	Urban	3	1.3	3.8	0.2	0.8
Centennial Brook	Urban	1	4.0	6.4	0.5	3.2
Centennial Brook	Urban	2	1.1	6.0	0.3	1.8
Englesby Brook	Urban	1	1.9	7.0	0.3	2.4
Indian Brook	Urban	1	19.6	8.6	0.4	3.1
Indian Brook	Urban	2	13.9	7.0	0.6	3.9
Morehouse Brook	Urban	1	1.1	4.3	0.2	0.8
Munroe Brook	Urban	1	13.9	6.1	0.4	2.5
Munroe Brook	Urban	2	6.1	8.4	0.5	4.1
Munroe Brook	Urban	3	3.7	4.5	0.2	1.0
Munroe Brook	Urban	4	4.6	3.5	0.3	1.0
Potash Brook	Urban	1	18.4	10.5	0.5	4.9
Potash Brook	Urban	2	18.2	8.5	0.5	4.5
Potash Brook	Urban	3	15.6	10.0	0.6	5.5
Potash Brook	Urban	4	11.3	6.7	0.4	2.7
Rugg Brook	Urban	1	7.4	4.8	0.3	1.5
Rugg Brook	Urban	2	6.8	6.3	0.4	2.3
Allen Brook	Rural	6	10.2	4.9	0.3	1.7
Allen Brook	Rural	7	4.1	4.0	0.3	1.3
Johnnie Brook	Rural	1	16.6	9.9	0.4	4.3
Johnnie Brook	Rural	2	16.4	10.9	0.5	5.9
Johnnie Brook	Rural	3	14.8	9.5	0.5	4.6
Johnnie Brook	Rural	4	10.0	4.7	0.3	1.4
Johnnie Brook	Rural	5	9.5	8.4	0.3	2.8
Johnnie Brook	Rural	6	6.0	4.8	0.5	2.2
Johnnie Brook	Rural	7	3.0	3.6	0.4	1.4
Johnnie Brook	Rural	8	1.3	2.5	0.3	0.7
Mill Brook	Rural	1	42.0	13.6	0.6	8.1
Mill Brook	Rural	2	36.3	16.9	0.6	9.9
Mill Brook	Rural	3	36.0	10.6	0.6	5.9
Mill Brook	Rural	4	35.1	11.7	0.5	5.8
Mill Brook	Rural	4	32.5	13.3	0.5	6.9
Mill Brook	Rural	5	30.8	10.7	0.6	6.6
Mill Brook	Rural	6	14.2	7.8	0.5	4.0
Sucker Brook	Rural	1	17.8	6.4	0.4	2.6
Sucker Brook	Rural	2	16.9	5.5	0.5	2.6

[1]Average of three to four bankfull values from each reach.

relationship of the ICM. Although a 10% TIA threshold has been discussed in many studies supporting the ICM, a recent review by Schueler *et al.* (2009) suggests that impacts from TIA may be detected with values as low as 4%. Other research by Walsh *et al.* (2005a) and Cuffney *et al.* (2010) suggests that impervious impacts to stream ecosystem health has no defined threshold of resistance. In addition, natural gradients and legacy effects may confound the relationship (Cuffney *et al.*, 2010), making it unlikely that a single threshold can be applied to different physiographic regions (Allan, 2004).

Our results using high-gradient reaches add to the growing number of results indicating that stream conditions may be impacted even at levels below 10% TIA in the upper Midwestern and New England regions (Morse, 2001; Stepenuck *et al.*, 2002; Wang and Kanehl, 2003; Schiff and Benoit, 2007). These results are generally characterized by some variability in biotic integrity and geomorphic stability at low levels of urbanization (<5% TIA), with a sharp decline between 5 and 10% TIA, followed by a leveled response in watersheds with TIA >10%. These results indicate that a decline in stream ecosystem health

TABLE 5. Spearman's Rank Correlations for Landscape Variables and Stream Physical and Biotic Indices.

	DA[1]	TIA[2]	Forest[3]	Slope[4]	% Cobble	% Sand	RGA	RHA	Richness[5]	EPT[5]	BI[5]
DA[1]	1										
TIA[2]	−0.48	1									
Forest[3]	**0.67**	**−0.80**	1								
Slope[4]	**−0.56**	0.12	−0.02	1							
% Cobble	0.03	−0.32	0.35	**0.53**	1						
% Sand	−0.27	0.26	**−0.60**	−0.36	**−0.74**	1					
RGA	0.40	**−0.90**	**0.75**	0.03	0.38	**−0.48**	1				
RHA	0.46	**−0.79**	**0.69**	−0.01	0.44	−0.43	**0.93**	1			
Richness[5]	**0.64**	−0.53	**0.83**	−0.04	0.36	**−0.58**	**0.55**	**0.45**	1		
EPT[5]	**0.63**	**−0.86**	**0.79**	−0.19	0.29	−0.52	**0.76**	**0.65**	**0.61**	1	
BI[5]	0.07	0.48	−0.17	0.03	−0.32	0.41	0.11	0.11	0.10	**−0.63**	1

Note: Bold entries are significant at $\alpha = 0.05$.
[1]Watershed drainage area.
[2]Total impervious area for upslope watershed.
[3]Watershed percent forest cover.
[4]Channel slope.
[5]Macroinvertebrate indices described in text.

will likely be detectable even at very low levels of urbanization (5-10% TIA). The increased sensitivity of stream conditions to low levels of urbanization is acknowledged in the reformulated ICM presented by Schueler *et al.* (2009).

The variability in stream ecosystem response below 5% TIA in New England is likely attributable to other legacy effects from humans (e.g., land clearing for agriculture) and natural watershed characteristics (e.g., wetlands, glacial history). The antecedent land use of an urbanizing watershed plays an important role in the future ecological condition of streams. Cuffney *et al.* (2010) showed a variable response across nine metropolitan areas in the U.S., illustrating the importance of the underlying natural (e.g., geology) and anthropogenic (e.g., historical land use) settings in each region. In our study area, where impacts to the infiltration capacity of the soils (e.g., reduced organic matter) have occurred and recurred over the last 200 years due to agriculture and forestry practices, it is possible that minor impacts from urbanization could lead to significant declines in stream condition.

The results of the macroinvertebrate metrics indicated that EPT and BI were negatively correlated as expected, and both EPT and species richness were negatively correlated with TIA as discussed previously. However, BI and TIA were not significantly correlated. The BI metric developed by Hilsenhoff (1987) is suited to detecting biotic degradation caused by organic and nutrient pollution typically associated with watersheds with a high degree of agricultural land use. This BI may have been less sensitive than the EPT metric to land use change because of the low levels of agricultural land use in our study watersheds.

Natural Gradients

Given the independent watershed approach and reach selection criteria used in this study, our results provide clear evidence that stream ecosystem health declines in response to urbanization in our study area as expected. However, we also observed very different physical responses between high- and low-gradient stream types. The decline of geomorphic stability in response to TIA was more significant and precipitous in high-gradient reaches than in low-gradient reaches. Differences in natural factors between the two stream types such as channel slope, floodplain width and connectivity, and channel boundary conditions (e.g., streambank soils and vegetation) likely explain the differences in response to upslope urbanization. Low-gradient stream types in our study area tend to have wider floodplains, greater sinuosity, and herbaceous streambank vegetation (due to recurring beaver activity). Greater dissipation of the erosive peak-flow forces caused by urbanization may be achieved by low-gradient streams that have adequate floodplain access and the ability to maintain and develop sinuosity. In addition, several studies have shown that stream channels with herbaceous bank vegetation tend to be narrower than those with woody vegetation (Davies-Colley, 1997; Hession *et al.*, 2003; Anderson *et al.*, 2004), a difference that may result from lower rates of erosion in channels with the former condition (Trimble, 1997).

The presence of beavers (*Castor canadensis*) in many of the low-gradient reaches in our study area made the assessment of physical channel conditions challenging and may have introduced additional variability and trends in our analysis. Through dam building and tree removal, beavers substantially

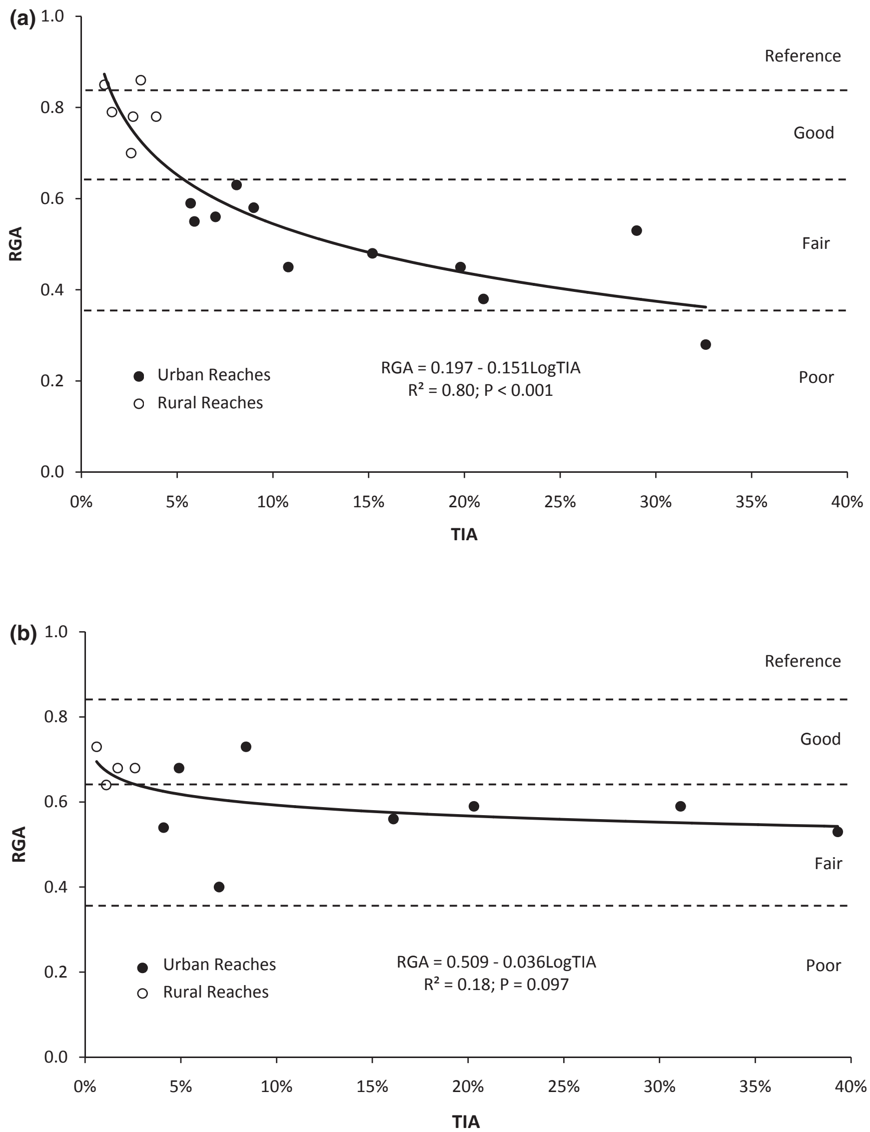

FIGURE 2. Plot of the Relationship Between RGA and Percent Upslope TIA for High-Gradient Study Reaches (a), and for Low-Gradient Study Reaches (b). Categorical groupings of physical stream condition provided on right (VTDEC, 2005).

impact the hydro-geomorphic characteristics of streams in our region in a way similar to boreal forests as described by Naiman *et al.* (1986). For ICM studies conducted in regions where beavers are abundant in low-gradient stream types, the analysis of stream response to urbanization may be improved by separating stream types based on gradient as we have done.

Our high-gradient stream results indicate that the response of stream condition depends on the interaction of other natural or inherent watershed characteristics with urban land cover. In our study watersheds, steeper headwater reaches appear to be more impacted by the effects of urban land cover than downstream, higher-order reaches. Other studies have reported that steep headwaters channels are most susceptible to rapid destabilization (Neller, 1989; Booth, 1990). Neller's (1989) research in Australia found that the magnitude of urban-induced channel enlargement in steeper, headwaters reaches was four to five times greater than lesser sloped, downstream reaches. In urbanizing watersheds in the Pacific Northwest, Booth (1990) noted that lower-order, higher-gradient stream reaches are most susceptible to rapid channel incision processes, and that high sheer stress in low-gradient, higher-order streams produced only minimal bed lowering.

Scale-Dependent Responses

Many studies have found significant correlations between local and upslope urban land cover (Fitzpa-

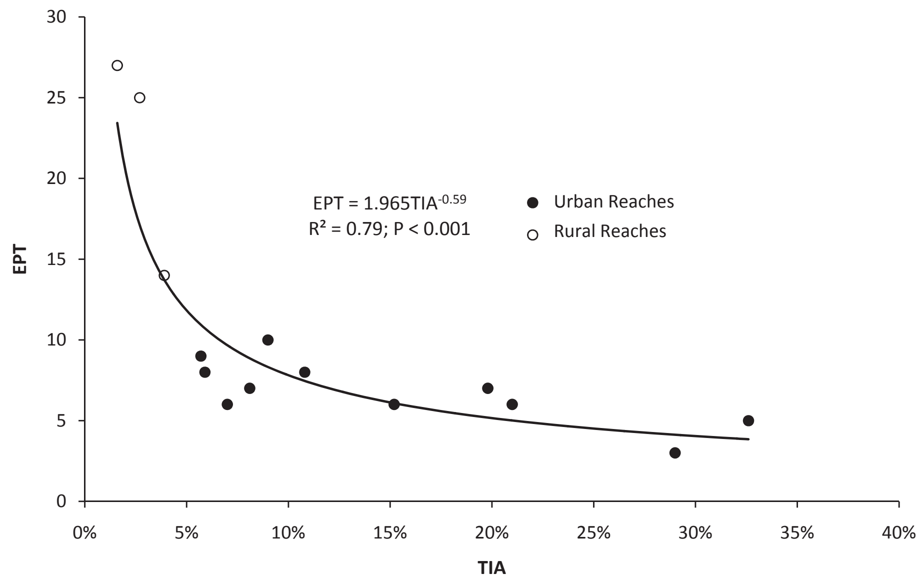

FIGURE 3. Plot of Relationship Between EPT Richness and Percent Upslope TIA for High-Gradient Study Reaches.

TABLE 6. Significant Variables in Stepwise Regression Analysis of RGA and EPT Richness for High-Gradient Reaches at Three Different Scales for TIA.

	RGA (n = 17)		EPT Richness (n = 14)	
Predictor Variables	**Coefficient**[1]	***p***	**Coefficient**[1]	***p***
Upslope area[2]	Model R^2 = 0.80		Model R^2 = 0.90	
TIA	−0.151	<0.001	−0.634	<0.001
% Cobble	N.S.	N.S.	0.017	0.004
Local area[2]	Model R^2 = 0.66		Model R^2 = 0.77	
TIA	−0.113	<0.001	−0.420	0.007
Drainage area[3]	N.S.	N.S.	0.173	0.080
% Sand	N.S.	N.S.	−0.014	0.071
Corridor[2]	Model R^2 = 0.52		Model R^2 = 0.55	
TIA	−0.050	0.005	N.S.	N.S.
Drainage area[3]	0.049	0.099	0.282	0.034
% Sand	N.S.	N.S.	−0.023	0.026

Notes: Model R^2, model adjusted R^2; NS, not significant in stepwise regression analysis and not included in model.
[1]Regression equation coefficient indicating the direction the variable relates to RGA and EPT richness.
[2]Scale at which percent TIA was measured (see Methods section).
[3]Watershed drainage area for all models.

trick *et al.*, 2001; Morley and Karr, 2002; Wang and Kanehl, 2003; McBride and Booth, 2005; Schiff and Benoit, 2007), making it difficult to conclude which scales are important for the management of TIA and its effects on stream conditions. In our study, percent TIA in the local area was highly correlated with percent TIA in the upslope area (ρ = 0.87, p < 0.001), as were corridor and upslope area TIA (ρ = 0.63, p = 0.006). Therefore, strong conclusions cannot be drawn about the importance of measuring localized TIA. However, our results generally agree with other urban watershed studies showing that localized and upslope urban land cover may both be important variables in predicting stream condition (Morley and Karr, 2002; Wang and Kanehl, 2003; McBride and Booth, 2005).

Our results indicate that the influence of urbanization on stream condition declines as urban land cover is measured at smaller spatial scales within the watershed. Models developed to predict RGA and EPT richness using percent TIA in the local area were also significant, but explained less variance

TABLE 7. Summary of Significant ANCOVA Results for ICM Analyses and DHG Regressions for High-Gradient Reaches.

DHG Regressions

Response Variable	Predictor Variables			
	Model	WT	DA	WT × DA
Width[1]	WT, DA, WT × DA	**0.001**	**<0.001**	**0.001**
Depth[1]	WT, DA, WT × DA	0.800	**<0.001**	0.453
Area[1]	WT, DA, WT × DA	**0.043**	**<0.001**	0.079

ICM Analyses

Response Variable	Predictor Variables					
	Model	WT	*S*	DA	WT × *S*	WT × DA
RGA	WT, *S*, WT × *S*	0.613	0.580	-	**0.011**	-
	WT, DA, WT × DA	**<0.001**	-	0.490	-	**0.037**

Notes: No significant interactions found for EPT models. Significant (α = 0.05) variables and interactions shown in bold. WT, watershed type (urban or rural); *S*, channel slope; DA, drainage area; -, not applicable.
[1]Bankfull dimensions.

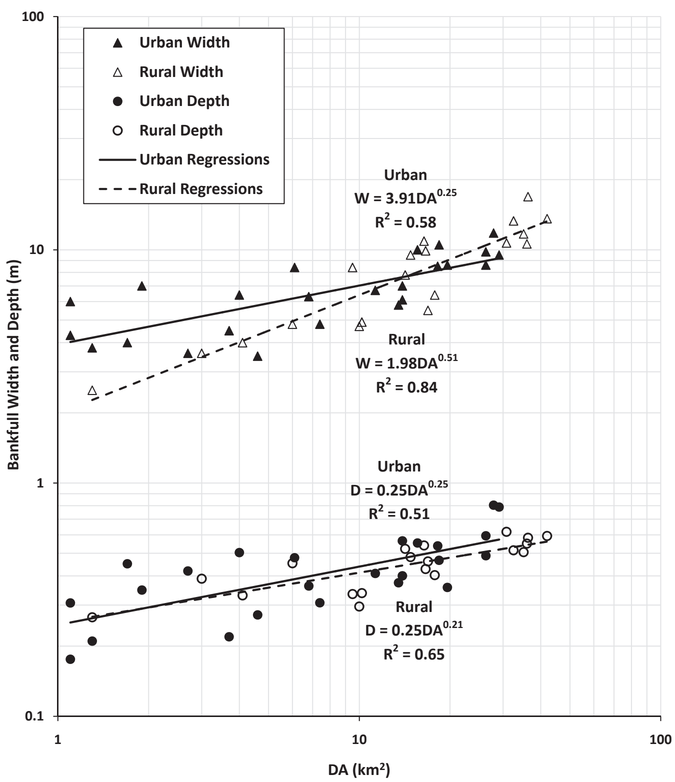

FIGURE 4. Plots of DHG Regressions for Bankfull Channel Width and Depth for Urban and Rural High-Gradient Stream Types. Regression equations noted on plot.

than upslope area models. However, unlike upslope area models, local area and corridor models of EPT were improved slightly when the DA variable was included with TIA (Table 6). This result suggests that although localized TIA may also have an effect on stream condition, this effect depends on the size of the entire watershed area draining to the reach. For example, in the case of a high-order reach located in the downstream area of a predominantly rural watershed, impervious area in the local area may not have a significant effect on stream hydrology (and associated stream processes), as its effect would become dampened by the natural watershed conditions upslope.

Results of the DHG regressions for high-gradient stream types indicate well-developed DHG relationships and provide strong evidence of a scale-dependent response of channel width to urbanization in Vermont (Figure 4). In reaches where DA is small (1-5 km^2), urban stream channels are wider than rural channels; however, this response diminishes as the DA approaches $\sim$15 km^2. The same response was observed for channel cross-sectional area, but not for channel depth. We explored possible explanations for the width and cross-sectional area response, including correlations between DA and TIA, and differences in channel slope between urban and rural stream types. We found no significant correlation between DA and TIA (Table 5), and no significant difference between the median value of the slope populations for the two stream types (α = 0.05).

Hession *et al.* (2003) noted the scale dependence of the channel-widening response to urbanization in the small watersheds they studied in the Piedmont region of the eastern U.S. In addition, several other studies from physiographic regions outside of the northeastern U.S. have shown a similar effect (Allen and Narramore, 1985; Doll *et al.*, 2002), with this effect typically diminishing between DAs of approximately 10 to 15 km^2. That a similar response has been observed across small watersheds with contrasting lithologies (Allen and Narramore, 1985) and vegetation types (Hession *et al.*, 2003) suggests that this phenomenon might be widespread in coarse-bottomed channels. Indeed, a comprehensive study examining DHG regressions across nine ecoregions of the U.S. (Faustini *et al.*, 2009) found that β values were lower in regressions developed for more disturbed watersheds than less disturbed watersheds in five of the nine ecoregions. It is possible that an interaction between the altered hydrologic regime and other inherent geomorphic characteristics that vary from headwaters to response zones, such as valley confinement, floodplain access, bed substrate, and channel sinuosity, could help explain the response. Channel evolution processes (Schumm *et al.*, 1984) may also advance at different rates due to a gradient in floodplain morphology across watershed scales. Booth (1990) found that lower-order urban stream reaches

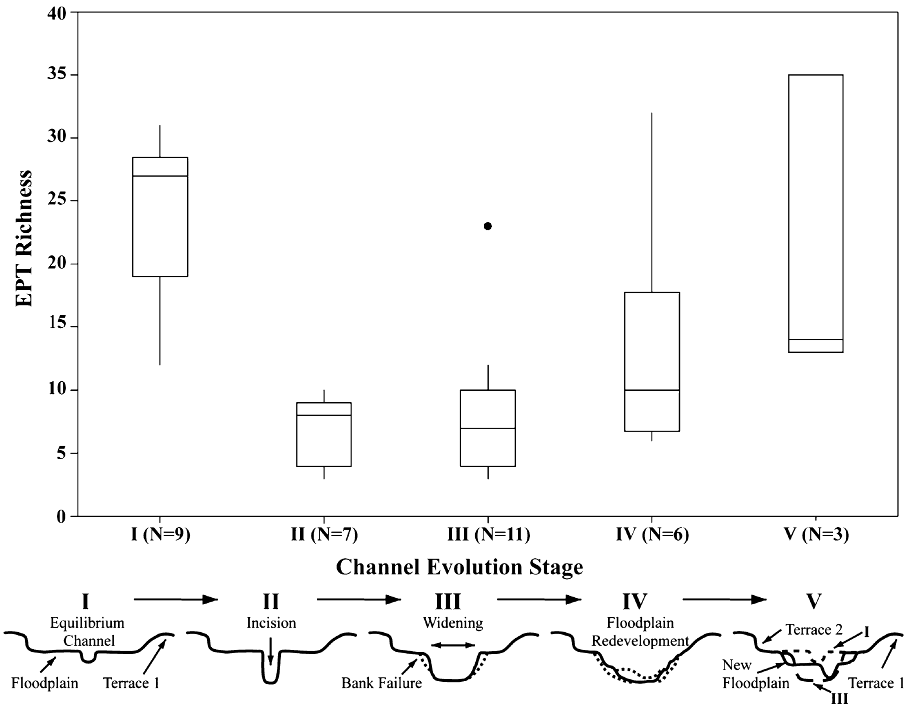

FIGURE 5. Boxplot of EPT Richness for Channel Evolution Stages in High-Gradient Stream Reaches (n = 36). Number in parentheses is macroinvertebrate sample size (i.e., number of sampled reaches) for channel evolution stage. Boxes represent first and third quartiles ($Q1$ and $Q3$), with the median value shown as the line between. Whiskers represent the range of data within limits 1.5 ($Q1$-$Q3$), with outliers represented by a dot. Channel cross-section diagram (bottom) depicts the channel adjustment sequence typically observed in urban watersheds (Schumm *et al.*, 1984; VTDEC, 2005). Unstable channel conditions are present in Stages II through IV, whereas stable conditions are present in the equilibrium (I) and quasi-equilibrium (V) stages.

are most susceptible to rapid channel incision processes than higher-order reaches. Further investigation of channel adjustment and evolution processes in urban streams is needed to understand the mechanics of this scale-dependent response.

Biotic Response to Channel Evolution Processes

Our results of macroinvertebrate response to channel evolution processes (Figure 5) suggest that there is a link between the spatial and temporal scales of geomorphic instability and biotic communities in our study watersheds. Increased stream power and sheer stress associated with the incision and widening stages (II and III, respectively) lead to more frequent scour of the stream bed and banks, resulting in unstable bedforms and fine sedimentation of the bed substrate. These channel adjustment processes, among many other effects of urbanization on stream ecosystems (Walsh *et al.*, 2005b), contribute to a loss of the quality habitat needed to support sensitive macroinvertebrate taxa. Stream power typically decreases as channels aggrade coarse substrate and redevelop sinuosity in Stage IV of channel evolution, and stream power in the quasi-equilibrium channel stage (V) is often similar to that observed in stable channels (Bledsoe *et al.*, 2002). Other studies in our region have linked geomorphic processes with biotic integrity in rural watersheds (Sullivan *et al.*, 2004, 2006); however, these results have been less conclusive for macroinvertebrates than fishes. Sullivan *et al.* (2004) carried out a study using RGA, RHA, and macroinvertebrate measurements from paired stable and unstable reaches in central Vermont. Although Sullivan *et al.* (2004) reported that stable reaches did not support significantly greater macroinvertebrate densities than unstable reaches, EPT richness was significantly correlated with geomorphic stability. Sullivan *et al.* (2006) also reported on the influence of geomorphic condition on fish communities in separate study sites in Vermont. Results from this study indicate that geomorphic stability was a significant predictor of fish communities, and suggest that fishes may be more responsive to geomorphic stability than macroinvertebrates due to differences in habitat scales.

A growing body of literature suggests that the maintenance of ecological integrity in stream ecosystems involves the complex interaction of many physical and biological processes at multiple spatial scales both within the channel (Thomson *et al.*, 2001; Poole, 2002) and in the stream corridor (Smith *et al.*, 2009). In the urbanized watersheds of our study area, increased runoff often results in channel incision and bed and bank instability. Once the channel incision process has begun, the loss of channel sinuosity reduces the potential surface area of habitat suitable for macroinvertebrate colonization. Narrowed, incised channels with homogeneous habitat support less diverse biotic communities than stable channels (Figure 5; Stages I and V) with geomorphic processes in or approaching a state of dynamic equilibrium. The typical response of biota to channel evolution processes in urban watersheds may be different from that observed in rural watersheds where other adjustment processes (e.g., aggradation of fine sediments from agricultural runoff) are more common. These differences in physical adjustment processes may explain why the relationships between geomorphic stability and biotic richness we report are stronger than those reported by Sullivan *et al.* (2004) from the same region.

SUMMARY AND CONCLUSIONS

This study provides additional evidence that relatively low levels of urbanization negatively impact stream conditions in the northeastern U.S. and that the response to this stressor can be measured using physical and biotic indices at multiple spatial scales. Our results also highlight the importance of considering watershed scale and other inherent characteristics in the response of stream conditions to urbanization. Headwaters areas with steep channel slopes are the most severely impacted zones in small urban watersheds in our study area. This knowledge may be critical for municipal and watershed planners concerned with the degradation of aquatic resources and can be incorporated into land use planning (e.g., zoning ordinances to control urbanization, stormwater and site design standards, river restoration and mitigation plans) to improve the protection of stream ecosystems.

DHG regressions comparing urban and rural watersheds proved useful in elucidating a scale-dependent response in our study area. This response, which appears consistent with DHG regression results from numerous other physiographic regions, has important implications for widespread practices of urban stream channel restoration. Because Rosgen (1996) suggested the use of DHG regressions as critical tools in stream channel design, numerous governmental agencies have adopted this approach and promoted the development of DHG relationships for this purpose. In many regions where urban stream restoration projects are desired for mitigation, limitations of stream discharge data preclude the development of DHG regressions using Q as the independent variable. In these situations, DHG regressions developed using DA as the independent variable could be misused if the concept of scale dependence is ignored. Failure to recognize scale-dependent watershed processes in the response of channel geometry to urbanization could lead to improperly scaled channel restoration designs and project failure.

Lastly, this study provides insight into the influence of channel stability on aquatic biota in small urban watersheds. Despite the abundance of data supporting the hypothesis that urbanization negatively affects stream ecosystems, fewer studies have explored whether biotic communities can improve or recover following natural or human-induced channel restoration in the urban environment (see review from Palmer *et al.*, 2010). Answers to this challenging question will become critically important as more resources are invested in urban channel restoration in the U.S. (Bernhardt *et al.*, 2005). Our results provide evidence that some improvement may be possible following natural restabilization or, perhaps, engineered restoration that mimics natural recovery of dynamic equilibrium conditions. Nevertheless, our evidence of recovery potential for urban streams should be taken in the context of several important caveats raised by others. Booth (2005) notes that there is limited evidence suggesting that the mitigation of nonhydrologic factors (e.g., channel geometry) can remediate the impacts of urbanization. Bernhardt and Palmer (2007) similarly argue that channel restoration approaches in urban settings must be considered within a broader context of watershed management approaches. Palmer *et al.* (2010) conclude that restoration programs aimed at improving instream habitat heterogeneity alone will not be successful; rather, watershed-scale approaches that recognize the interactions between stressors at various spatial scales are needed. Channel evolution adjustments, the natural maintenance and redevelopment of channel habitat features, and the subsequent response of aquatic biota to these processes are directly tied to the altered sediment and hydrologic regimes brought on by urbanization. Therefore, long-term remediation efforts in urban-impaired watersheds will need to address these altered regimes at the watershed scale and protect lands within the stream corridor to provide the lateral space required for streams to redevelop equilibrium forms (Kline and Cahoon, 2010).

SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article.

Table S1. Reach Geomorphic and Biotic Assessment Results for Additional CEM Study Reaches.

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ACKNOWLEDGMENTS

This research was supported in part by EPA Section 319-01 funding directed through VTDEC and an Undergraduate Research Endeavors Competitive Award from the University of Vermont. The VTDEC River Management and Biomonitoring staff assisted throughout the data collection effort. D. Rizzo and M. Watzin provided valuable insights and reviews during the analysis. J. Foley, D. Lefevre, J. Carrano, R. Balouskus, and L. Moore provided essential assistance with the field work. M. Winslow and the Lake Champlain Committee shared data collected on Rugg and Stevens Brooks. This paper was improved by the thoughtful comments and suggestions from three anonymous reviewers.

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